

10-3-2007

Interactions Across Spatial Scales among Forest Dieback, Fire, and Erosion in Northern New Mexico Landscapes

Craig D. Allen

US Geological Survey, Jemez Mountains Field Station, Los Alamos, New Mexico

Follow this and additional works at: <http://digitalcommons.unl.edu/natrespapers>



Part of the [Natural Resources and Conservation Commons](#)

Allen, Craig D., "Interactions Across Spatial Scales among Forest Dieback, Fire, and Erosion in Northern New Mexico Landscapes" (2007). *Papers in Natural Resources*. Paper 104.

<http://digitalcommons.unl.edu/natrespapers/104>

This Article is brought to you for free and open access by the Natural Resources, School of at DigitalCommons@University of Nebraska - Lincoln. It has been accepted for inclusion in Papers in Natural Resources by an authorized administrator of DigitalCommons@University of Nebraska - Lincoln.

Interactions Across Spatial Scales among Forest Dieback, Fire, and Erosion in Northern New Mexico Landscapes

Craig D. Allen*

US Geological Survey, Jemez Mountains Field Station, Los Alamos, New Mexico 87544, USA

ABSTRACT

Ecosystem patterns and disturbance processes at one spatial scale often interact with processes at another scale, and the result of such cross-scale interactions can be nonlinear dynamics with thresholds. Examples of cross-scale pattern-process relationships and interactions among forest dieback, fire, and erosion are illustrated from northern New Mexico (USA) landscapes, where long-term studies have recently documented all of these disturbance processes. For example, environmental stress, operating on individual trees, can cause tree death that is amplified by insect mortality agents to propagate to patch and then landscape or even regional-scale forest dieback. Severe drought and unusual warmth in the southwestern USA since the late 1990s apparently exceeded species-specific physiological thresholds for multiple tree species, resulting in substantial vegetation mortality across millions of hectares of woodlands and forests in recent years. Predictions of forest dieback across spatial scales are constrained by uncertainties associated with: limited knowledge of species-specific physiological thresholds; individual and site-specific variation in these mortality thresholds; and positive feedback loops between rapidly-responding insect herbivore populations and their stressed plant hosts, sometimes resulting in nonlinear “pest” outbreak dynamics. Fire behavior also exhibits nonlinearities across spatial scales, illus-

trated by changes in historic fire regimes where patch-scale grazing disturbance led to regional-scale collapse of surface fire activity and subsequent recent increases in the scale of extreme fire events in New Mexico. Vegetation dieback interacts with fire activity by modifying fuel amounts and configurations at multiple spatial scales. Runoff and erosion processes are also subject to scale-dependent threshold behaviors, exemplified by ecohydrological work in semiarid New Mexico watersheds showing how declines in ground surface cover lead to non-linear increases in bare patch connectivity and thereby accelerated runoff and erosion at hillslope and watershed scales. Vegetation dieback, grazing, and fire can change land surface properties and cross-scale hydrologic connectivities, directly altering ecohydrological patterns of runoff and erosion. The interactions among disturbance processes across spatial scales can be key drivers in ecosystem dynamics, as illustrated by these studies of recent landscape changes in northern New Mexico. To better anticipate and mitigate accelerating human impacts to the planetary ecosystem at all spatial scales, improvements are needed in our conceptual and quantitative understanding of cross-scale interactions among disturbance processes.

Key words: disturbance interactions; forest dieback; fire; erosion; fire history; cross-scale relationships; thresholds; New Mexico; southwestern USA.

Received 11 September 2006; accepted 10 April 2007; published online 20 June 2007.

*Corresponding author; e-mail: craig_allen@usgs.gov

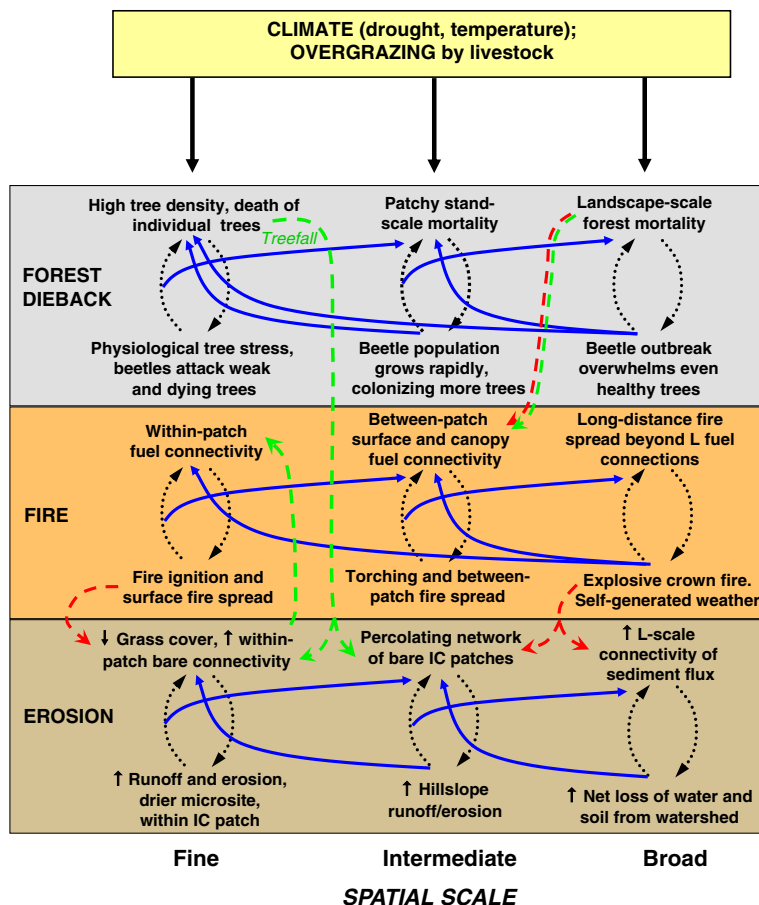


Figure 1. Diagram representing interactions across spatial scales for three different disturbance processes (forest dieback, fire, erosion) in northern New Mexico landscapes, building upon Figure 1(a) in Peters and others (2007). *Dashed black arrows* represent pattern-process feedbacks within three different spatial scale domains, with one example of pattern and process shown for each domain for each disturbance. *Solid black arrows* indicate the overarching direct effects of widespread environmental drivers or disturbances (climate, overgrazing) on patterns and processes at all scales. *Blue arrows* indicate the point at which altered feedbacks at finer spatial scales induce changes in feedbacks at broader scales (for example, fine-scale changes cascade to broader scales), and also where changes at broader scales overwhelm pattern-process relationships at finer scales. *Red dashed arrows* illustrate some text examples of amplifying (positive feedback) interactions between disturbance processes, within and between spatial scales; *green dashed arrows* illustrate dampening (negative feedback) interactions between disturbance processes (between disturbance processes, within and between spatial scales; *L* landscape; *IC* intercanopy (interspaces between tree canopies)).

INTRODUCTION

Ecosystem patterns and disturbance processes at one spatial scale often interact with processes at another scale, and the result of such cross-scale interactions can be nonlinear dynamics with thresholds (Gunderson and Holling 2002). Such interactions change the pattern-process relationships across scales such that fine-scale processes can influence a broad spatial extent, or broad-scale drivers can interact with fine-scale processes to determine system dynamics (Peters and others 2007). Non-linear disturbance propagation and amplification across spatial scales commonly involve positive feedback loops or spatial pattern-process thresholds, differing from linear relationships observed with simple repetition of patterns from fine to broad scales. Cross-scale interactions involving disturbance processes are increasingly recognized as important drivers of ecosystem dynamics (Nepstad and others 2001; Turner 2005; Burkett and others 2005; Bodin and others 2006; Peters and others 2007). Spatially explicit approaches differ from non-spatial approaches in

their emphasis on the importance of the interaction of disturbance transfer processes with spatial heterogeneity (Peters and others 2007). This paper illustrates examples of nonlinear interactions across spatial scales among forest dieback, fire, and erosion from landscapes in the southwestern USA (Figure 1), particularly northern New Mexico, where long-term studies centered on Bandelier National Monument in the Jemez Mountains have recently documented all of these disturbance processes (Allen 1989).

Methods used to develop the presented examples include: dendrochronological dating of fire scars (Touchan and others 1996) and tree population demographics (Swetnam and Betancourt 1998), weekly dendrometer measurements of ponderosa pine (*Pinus ponderosa*) and piñon (*Pinus edulis*) tree-growth since 1991 and 1993 (CD Allen, unpublished data), monitoring of basal cover of herbaceous vegetation at centimeter resolution along 3 km of permanently marked line-intercept transects since 1991 (CD Allen, unpublished data), photographic and remote-sensing delineation of forest dieback (Allen and Breshears 1998; Bres-

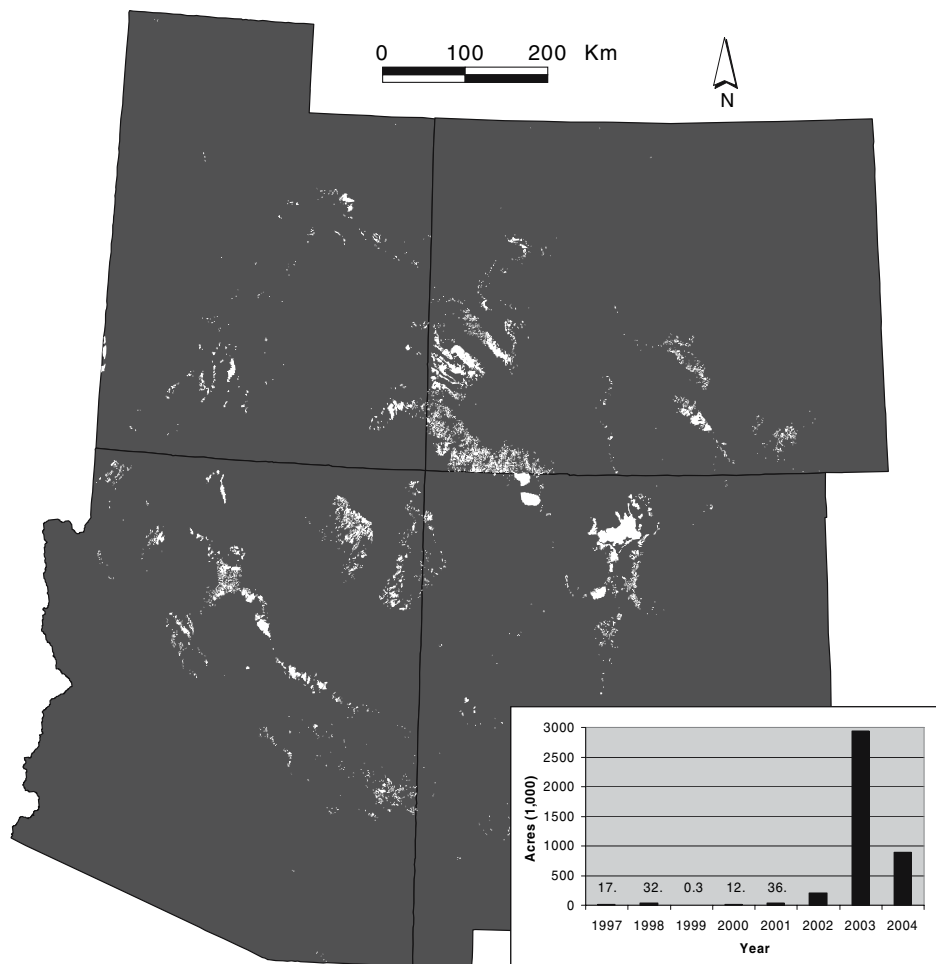


Figure 2. Extent of *Pinus edulis* dieback from 1997 to 2004 in the Four Corners states of Arizona, New Mexico, Colorado, and Utah. Based upon annual aerial insect pest inventories by the US Forest Service.

hears and others 2005), aerial surveys of forest insect pest activity (US Forest Service 2003), and detailed multi-scale measurements of runoff and erosion since 1993 (Wilcox and others 2003). These diverse methods provide multiple perspectives on ecosystem patterns and processes across spatial scales in northern New Mexico landscapes.

INTERACTIONS ACROSS SPATIAL SCALES

Forest Dieback and Associated Insect Outbreaks

Environmental stress, operating on individual trees, can directly cause tree death that accumulates in linear fashion at broader spatial scales (Figure 1 black climate arrows) or that interacts with other disturbance processes to propagate nonlinearly to patch and then landscape or even regional-scale forest dieback (Figure 1 dieback panel). Because of interactions between mortality-related disturbance processes across spatial scales, forest dieback may not

be attributable to a single cause. For example, vegetation dieback (and associated insect outbreaks) have affected millions of hectares of forests and woodlands in the Southwest since the late 1990s (Figure 2), a period of severe drought and unusual warmth (US Forest Service 2003; Breshears and others 2005). The combination of extreme drought and warmth apparently exceeded species-specific physiological thresholds of mortality for multiple tree species (Breshears and others 2005; Burkett and others 2005; Gitlin and others 2006). Although direct climate-induced stress may have been sufficient to have caused recent regional-scale forest dieback, substantial insect pest populations built up in weak and dying trees, confounding the underlying and ultimate causes of extensive tree mortality patterns. Note that recently in the Southwest substantial climate-induced mortality also has affected non-tree plant lifeforms ranging from cacti to shrubs and grasses (personal observation, and see DIRENET website at <http://www.mpcer.nau.edu/direnet/overview.html>), without the apparent involvement

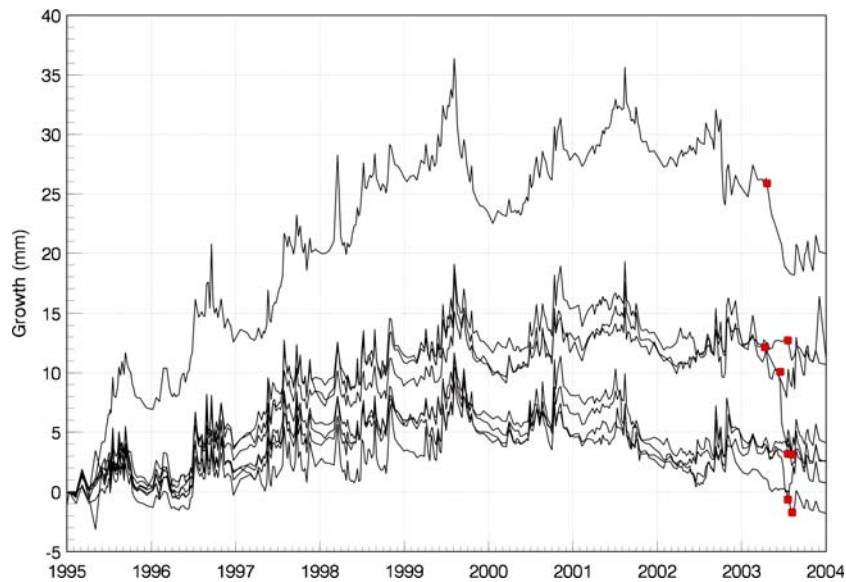


Figure 3. Cumulative growth (change in circumference) of eight piñon trees at Bandelier National Monument, 1995–2003, including shrinkage through dehydration during dry periods. Net growth ceased in about 2000 due to drought-related physiological stress, and as the drought persisted and intensified into 2002 these piñon all eventually died during 2003 (red squares mark death dates).

of significant biotic mortality agents. For example, repeated measurements along 2,700 m of permanent vegetation transects in piñon-juniper woodlands at Bandelier National Monument in northern New Mexico revealed reductions between 1999 and 2003 in the live basal cover of the dominant perennial grass (*Bouteloua gracilis*), from 2.5% to less than 1%, apparently due to drought-induced mortality (CD Allen, unpublished data). McAuliffe and others (2006) found similar patterns of grass death between 1999 and 2002 in northeastern Arizona, and Mirit and others (2007) document 55–100% adult mortality of five dominant shrub species at a Mojave Desert site due to the regionally severe drought in 2002.

The recent regional-scale dieback of piñon in the American Southwest (Figure 2) specifically illustrates the propagation of tree death across spatial scales, driven by positive feedbacks between drought stress and mortality-causing insect populations (Figure 1). Measurements of piñon tree growth (Figure 3) reveal that drought-induced stress constrained tree growth for several years, driving a “death spiral” (Franklin and others 1987; Pederson 1998) by which all of these weakened trees died in 2003. The death of individuals propagated up to stand and landscape scales (Figure 1), in part through amplification of mortality by an associated outbreak of *Ips confusus*, a bark beetle (Shaw and others 2005). These beetles infest piñon through mass attacks that can overcome the diminished pitch defenses of weakened trees, burrowing into their stems and laying approximately 60 eggs per female that become larvae that burrow through the tree’s cambium, girdling and killing

the tree. In the southwestern USA these piñon *Ips* generally have two generations per year, permitting in theory an annual population increase of approximately two orders in magnitude. As the defenses of individual drought-stressed trees failed, beetle populations built up rapidly, leading to a coupled positive feedback whereby growing numbers of beetles became available to attack increasingly weak trees, resulting in population irruptions with great abundances of beetles that could fly to adjoining areas and successfully infest even relatively robust piñon trees. The synergistic result of the warm drought and associated beetle outbreak in 2002–2004 was wholesale mortality of piñon at stand, landscape, and regional scales. For example, monitoring of piñon demography in two 0.5 ha permanent plots at Bandelier National Monument shows that more than 95% of all non-seedling piñon died during 2002–2003, including all trees greater than 5 cm in basal diameter and 2 m tall (CD Allen unpublished data), with over 90% of mature piñon individuals killed at landscape scales across much of the Jemez Mountains, and more than 1,000,000 ha of substantial piñon dieback across the region (Figure 2, US Forest Service 2003; Breshears and others 2005). Protracted and severe drought in the Southwest led to spatial synchrony of this mortality, by entraining piñon populations in climate-driven stress across the region. Such massive dieback of a dominant species can have long-term and wide-ranging ecological effects at local, landscape, and even regional scales (Mueller and others 2005).

Although the basic driver of the recent vegetation dieback in the Southwest has been climate-

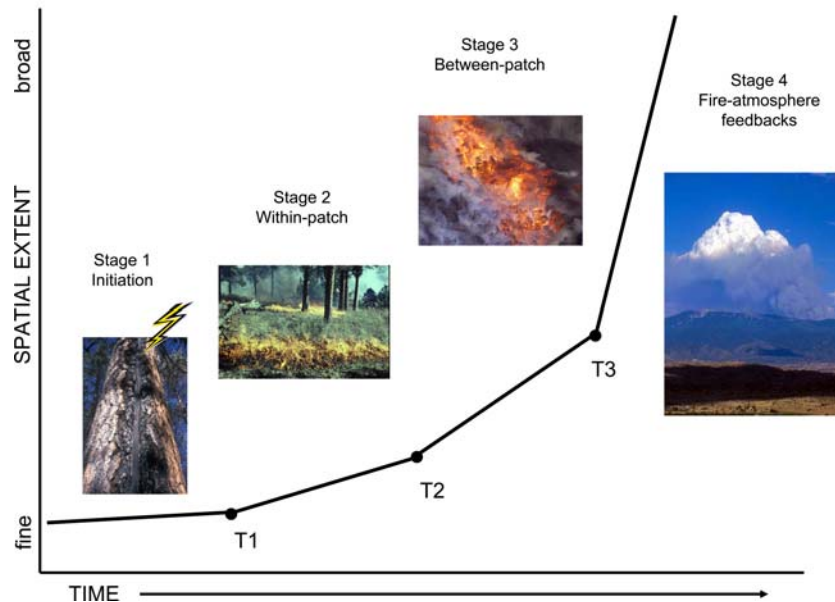


Figure 4. Spatial nonlinearities and three thresholds (T1–T3) through time for wildfire, adapted from Figure 1 in Peters and others (2004). Stage 1 represents fire initiation from a lightning strike. Fire spread rate increases in Stage 2 if within-patch fuel continuity is adequate. If the spatial distribution and connectivity of fuels support fire spread between patches, then fire behavior can transition through T2 to Stage 3. At the broadest spatial scale, past T3 into Stage 4, positive feedbacks between the atmosphere and the fire can generate extreme, rapidly spreading fire behavior that can transcend fuel connectivity within or between patches. Stage 4 fire often generates pyrocumulus clouds, as pictured here with the Viveash Fire of 2000 in northern New Mexico.

induced stress, associated outbreaks of insects such as bark beetles can greatly amplify woody plant mortality when insect population growth escapes from local controls through positive feedbacks to propagate into broad-scale irruptions (Figure 1). The threshold influences of climate in triggering large-scale insect outbreaks (for example, by modifying life cycle dynamics and food resource availabilities across spatial scales) are addressed in a number of studies, including Ludwig and others (1978), Berryman and others (1982), Logan and Powell (2001), Powell and Logan (2005), and Sanfranyik and Carroll (2006).

Climate change models predict substantial shifts in climate during coming decades in many regions, including warmer temperatures and increases in extreme drought events (IPCC 2001). Such changes are likely to increase stress on long-lived woody vegetation, directly leading to episodes of increased mortality and forest dieback, as seen in past droughts (Allen and Breshears 1998). Forest stress and dieback are now becoming apparent in many parts of the world (Fensham and Holman 1999; Gonzales 2001; Suarez and others 2004; Breshears and others 2005). The potential for extensive climate-induced vegetation mortality is a key research topic, because mortality losses of woody species can occur much faster than replacement by

tree growth, with pervasive and persistent ecological effects. The effects of rapid forest die-off range from feedbacks to other disturbance processes such as fire and erosion (see text below and Figure 1) to loss of sequestered carbon back to the atmosphere (Breshears and Allen 2002). Cross-scale predictions of climate-induced vegetation dieback are constrained by uncertainties associated with: limited knowledge of species-specific physiological thresholds of plant mortality; individual and site-specific variation in these mortality thresholds; and coupled positive feedbacks between climate, host plant stress, and insect “pest” outbreak dynamics.

Wildfire Propagation

Wildfire behavior exhibits nonlinear pattern-process relationships across spatial scales, illustrated by changes in historic fire regimes and recent extreme fire events in New Mexico. Peters and others (2004) present a general framework of spatial nonlinearities, describing the propagation of changes in a phenomenon across spatial scales. Figure 4 shows this framework for wildfire, illustrating spatial nonlinearities and three thresholds (T1–T3) through time (adapted from and described in greater detail in Peters and others 2004). Stage 1 represents fire initiation from a lightning strike.

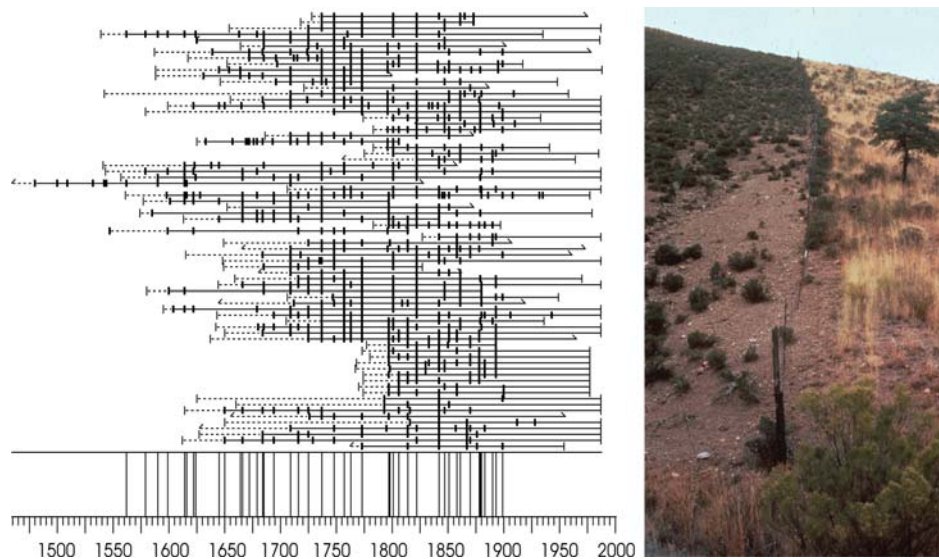


Figure 5. Composite fire scar chronology from the Rito de los Frijoles watershed in the Jemez Mountains, New Mexico (Allen 1989). Horizontal lines show the calendar-year life spans of individual trees, and the short vertical lines are fire scar dates. The longer vertical lines at the bottom of the chronology indicate years with spreading fires, in which fire scarred at least 10% of previously scarred trees and a minimum of two samples. The sampled trees are arrayed from lowest elevation (bottom, 2016 m) to highest (top, 2934 m). Note the high frequency of fire activity prior to 1900, the synchronism of fire years across samples, and the cessation of spreading fires after about 1900. The fence-line photograph shows the effect of intensive cattle grazing on grass cover (left side), reducing the continuity of surface fuels that sustain the propagation and spread of fire while increasing the connectivity of bare soil patches.

Fire will spread within the local patch and start to torch into canopies in Stage 2 if within-patch fuel continuity (horizontal and vertical) is adequate (Whelan 1995; Cram and others 2006). Note that fuel connectivity is partly a function of climate-driven moisture conditions that determine whether potential fuels (live and dead biomass) are sufficiently dry to be “available” to burn. If the spatial distribution and connectivity of available fuels support fire spread between patches, then the fire behavior can transition to Stage 3 (compare Miller and Urban 2000; Finney and others 2006). At these fine (within-patch) and intermediate (between-patch) scales, fire propagation depends upon the spatial pattern of fuels being sufficiently connected, that is, above a percolation threshold (Turner and others 2001). However, at the broadest spatial scale, in Stage 4, positive feedbacks between the atmosphere and the fire may generate extreme, rapidly spreading crown fire behavior (including long-distance fire propagation by “spotting”) that does not depend upon direct fuel connectivity within or among the patches. The positive feedback emerges from fire-created weather, in which the heat of the fire modifies local winds and humidity, allowing the process of fire to escape the constraining patterns of local fuel flammability and connectivity to affect a broad area, further gener-

ating even more heat and explosively spreading fire activity.

A complementary representation of cross-scale wildfire propagation is provided in Figure 1, with threshold T2 (unlabeled) analogously located in the transition between fine and intermediate spatial scales, and T3 between intermediate and broad scales. The escape of fine-scale pattern to generate broad-scale process in Stage 4 crown fire corresponds to the irruption stage of a bark beetle outbreak, as large exponentially-growing beetle populations are able to attack and kill even healthy and/or distant trees (compare Figure 1 broad-scale column).

Changes in the historic fire regimes of southwestern ponderosa pine forests show the role of cross-scale nonlinearities in determining wildfire behavior. Fire scar studies (Figure 5) reveal that prior to approximately 1900, southwestern ponderosa pine forests were subject to widespread and frequent surface fires [approximately 5–18 years between spreading fires at the scale of individual stands (Swetnam and Baisan 1996; Allen 2002)]. For example, fire scars sampled across a 20 km-long stretch of the Rito de los Frijoles watershed in the Jemez Mountains of northern New Mexico (Figures 5, 6) show extensive fire activity at about 5–16 years intervals from CE 1598 to 1899 (Allen

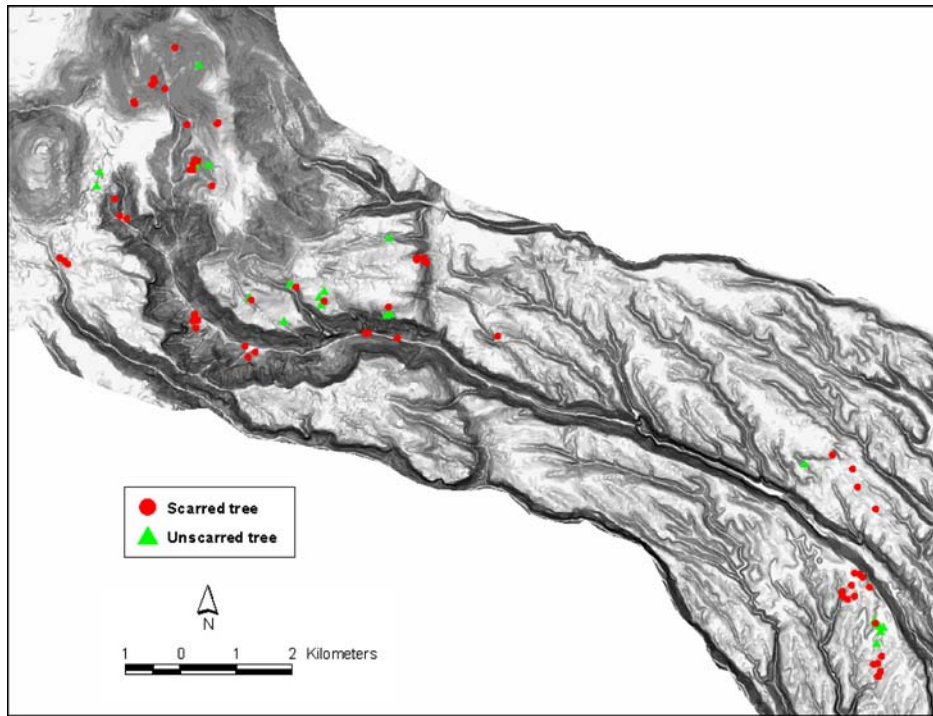


Figure 6. Map of the Rito de los Frijoles watershed with locations of sampled trees scarred by fire in 1773 (Allen 1989 and unpublished data). Three-meter contour intervals are displayed, so dark tones represent steep slopes. Map elevations range from 3,111 m at upper left to 1,680 m at lower right. Watershed-wide fire activity commonly occurred across diverse topographic and elevational settings, spanning mountain slopes and upland mesas to the bottom of 300-m deep Frijoles Canyon.

1989; Touchan and others 1996), with surface fires spreading across the majority of this basin approximately every 16 years on average (Allen 1989). The spatial pattern of watershed-scale fire occurrence indicates that fire likely propagated long distances across at least a 1,000-m elevational gradient that includes rugged terrain (Figure 6). Surface fuels (grass, needle litter) were sufficiently continuous to allow fires to spread widely in these and similar forests. Fire scar records also show that in particular years extensive surface fires affected much of the Jemez Mountains landscape (Allen 2002). Further, in certain years widespread fire activity occurred in most mountain ranges across the Southwest (Swetnam and others 1999), apparently dwarfing the size of any single recorded modern fire. These regional-scale fire years were synchronized by interactions between climate and fuels (Swetnam and Betancourt 1998), as the most extensive fire activity occurred in dry years that followed grass-growing wet years, fostering surface fuel continuities and thus better surface fire propagation and spread (Touchan and others 1996; Swetnam and Baisan 2003). Yet these spreading surface fires generally ceased by approximately CE 1900 (Figure 5), several decades before active and effective fire suppression by land management agencies. Why?

The sensitivity of pre-1900 surface fire regimes in the Southwest to land use and climate-driven changes in fuel abundance and connectivity

(Swetnam and Baisan 2003) indicates that the surface fuels in ponderosa pine ecosystems were often near thresholds of continuity for sustaining widely propagating fire. In the case of northern New Mexico, sustained disruption of the surface fuel patterns is associated with the entry of railroads in 1880 that provided access to external markets and investment capital into this formerly economically isolated region (Rothman 1992), driving a huge boom in livestock numbers (Wootton 1908). Extreme year-round overgrazing by domestic livestock then occurred from about 1880–1935 (Allen 1989), directly interrupting the spatial continuity of the fire-propagating matrix of herbaceous surface fuels in these open forests through herbivory and trailing (Figure 5). As surface fuels dropped below connectivity thresholds, fire spread within and between patches was markedly constrained (Figure 4 Stages 2 and 3), and surface fire regimes collapsed across multiple spatial scales, from local to regional (Swetnam and others 1999; Allen 2002). In this case, overgrazing restructured the pattern of fuels in the historical pattern-process linkage between surface fuels and surface fire, with consequent cross-scale transformation of the associated process of fire (Figure 1).

This regional-scale disruption of surface fire regimes subsequently led to an additional coupled feedback—without the tree-thinning effects provided by previously frequent surface fires, high densities of young ponderosa pine (as well as less

fire-resistant species like Douglas-fir) established in southwestern pine stands, profoundly further changing the ecological and fuel structures of these forests (Moore and others 1999; Allen and others 2002). This “woodification” of southwestern ponderosa pine forests increased the horizontal and vertical connectivity of the woody fuels, initially within individual forest patches, but eventually these dense forest patches became connected and coalesced at landscape scales. Fire behavior has changed accordingly (Figure 1), highlighted by increasingly frequent and large high-severity ponderosa pine crown fires in recent decades (Allen and others 2002). These extensive stand-replacing fires are in many cases resulting in “type conversions” from ponderosa pine forest to other physiognomic types (for example, grassland or shrubland) that may be persistent for centuries or perhaps even millennia (Savage and Mast 2005). Recent management emphases on ecological restoration and hazard fuel reduction in southwestern ponderosa pine forests increasingly include strategic efforts to interrupt landscape-scale fuel continuities with the placement of thinning and burning treatments (Finney 2001; Allen and others 2002; Sisk and others 2005).

Although these changes in pattern-process relationships across spatial scales between fuels and fire behavior have been essential in driving changes in southwestern fire regimes over the past 120 years, climate variability and change are also important (Swetnam and Betancourt 1998; Kitzeberger and others 2007). The onset of persistent drought and warm temperatures in the Southwest since about 1996, in combination with the extensive landscape scale of contiguously overgrown woody fuels, resulted in the largest fires in recorded history in Arizona (Rodeo-Chediski Fire, 189,798 ha), New Mexico (Ponil Complex Fire, 37,260 ha), and Colorado (Hayman Fire, 55,080 ha) during the extreme drought year of 2002. Interactions among drought and warmth, fuels, and fire behavior have fostered increasingly extensive and severe fire activity in the western USA (Westerling and others 2006), including the Southwest. Given climate change projections of greater warmth and drought for this region (Seager 2007), further transformations in the particulars of cross-scale fuel-fire relationships can be expected.

Massive invasions of exotic grasses (particularly *Bromus* and *Schismus* spp.) and forbs (for example, *Sisymbrium* and *Brassica* spp.) into arid lands of the western USA have increased the connectivity of surface fuels above fire transmission thresholds, triggering marked increases in the spatial scale of

fire activity in these formerly fuel-limited systems (Knapp 1998; Brooks and others 2004). This is apparent in USA wildland fire statistics, as outside of Alaska most of the large fires (>100,000 acres) since 1997 (see http://www.nifc.gov/stats/lrg_fires.html) are associated with increases in surface-fuel continuity in the Great Basin, Mojave Desert, and Sonoran Desert (for example, Brooks and Matchett 2006). For example, in 2005 the Cave Creek Complex Fire burned 100,566 ha of Arizona desert and foothills, and the Southern Nevada Complex fire burned 298,658 ha, the largest fire complexes ever recorded in either desert region. Once initiated, this pattern-process relationship of pyrophyllitic invasive grass spread and increased fire activity can enter a positive feedback cycle with the potential to substantially transform western deserts into alien ecosystems (D’Antonio and Vitousek 1992; Esque and Schwalbe 2002; Brooks and others 2004), as the native biota is generally poorly adapted to frequent fire whereas the area dominated by the invasives tends to ratchet upward after each fire. This increased fire activity in desert basins also spreads upslope into adjoining lower mountain slopes, changing the fire regimes of shrublands, woodlands, and forests (Brooks and Matchett 2006).

Linked Effects of Vegetation Surface Cover on Fire and Erosion Processes

Livestock overgrazing in the Southwest historically acted as a disturbance across spatial scales by directly altering the density and spatial patterning of herbaceous land cover, thereby modifying both fire regimes and erosion processes (Figures 1, Figure 7). Where herbaceous ground cover is sufficiently continuous to allow surface fires to spread (Figure 7a), it is also sufficient to retard the overland flow of water, fostering infiltration and increases in plant-available water (Davenport and others 1998; Wilcox and others 2003). Reductions in surface cover of plant materials (like grass and litter) below critical threshold values by grazing or drought-induced dieback can cause surface fire activity to collapse (Figure 1 green arrow from erosion to fire, Figure 7). Such reductions in vegetation cover also can trigger nonlinear increases in erosion rates as isolated bare soil patches become connected, coalescing into percolating networks at broader spatial scales (Figure 1 erosion panel, Figure 7b) that promote accelerated water runoff and associated erosion (Davenport and others 1998; Wilcox and others 2003). The increased net losses of water and soil feed back to reduce the productivity and vigor

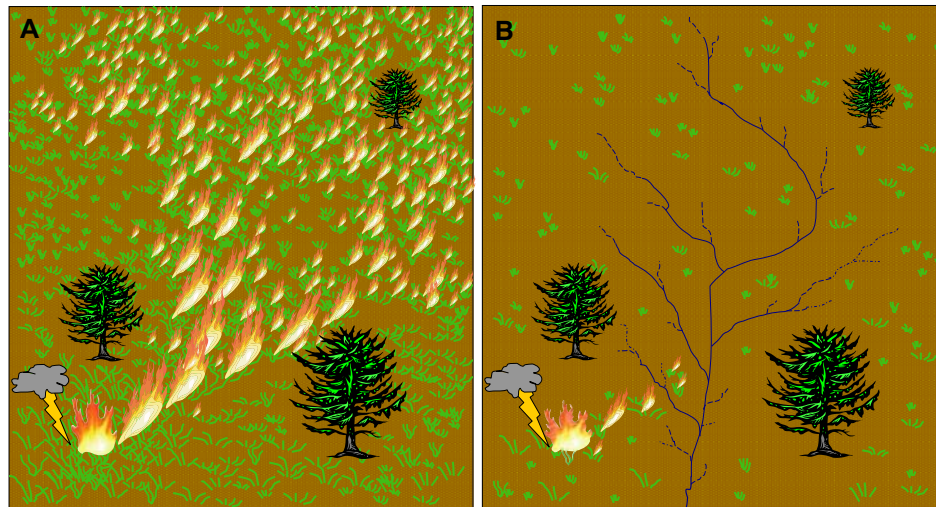


Figure 7. Schematic diagram of the essential role of herbaceous ground cover (shown as green grass clumps) in determining patterns of surface fire spread and water runoff and erosion. In (a) herbaceous cover is sufficiently dense and connected to allow wind-driven surface fire to spread across this landscape, as well as constrain soil erosion. In (b) major reductions in grass cover drop the continuity of the surface fuels below a critical threshold of connectivity, greatly constraining the ability of surface fire to spread. Conversely, the reduced grass cover results in greater connectivity of bare soil patches, exceeding a threshold that flips this ecosystem into a state of accelerated runoff and erosion, represented by the emergent channel network.

of vegetation cover, generating a desertification spiral (Figure 1 erosion panel, blue feedback arrows from broader to finer scales; Schlesinger and others 1990). Once initiated, altered pattern-process relationships of accelerated erosion may persist for decades (Wilcox and others 2003), and once desertified through loss of vegetation and soils, semi-arid ecosystems may be slow to recover (Peters and others 2006).

Similarly, rapid and extensive changes in watershed ecohydrology often occur when high-severity fires initiate nonlinear amplification of runoff and erosion by reducing vegetation and ground cover across broad areas (Shakesby and Doerr 2006), thereby triggering the development of highly connected water-shedding surfaces (Figure 1 red arrows from fire to erosion). The threshold hydrologic effects of such fire-induced land-cover changes are demonstrated by the approximately 100-fold increases in peak runoff observed for 1–3 years after large stand-replacing fires in the Jemez Mountains of New Mexico (Johansen and others 2001; Veenhuis 2002).

Disturbance Interactions Within and Across Spatial Scales

Multiple disturbance processes interact within and across spatial scales through time, including both positive and negative feedbacks, sometimes with

non-linear threshold effects. As a result of these synergistic complexities, disentangling the specifics of cross-scale disturbance interactions on the ecology of a particular landscape can be challenging. Several examples of disturbance interactions in southwestern landscapes have been described in this paper (Figure 1), including: (1) the role of climate in triggering both forest dieback and erosion through drought stress on dominant vegetation (trees and grass); (2) climatic extremes (drought, high temperature) as drivers of severe and extensive fire activity; (3) relationships between overgrazing, vegetative ground cover, surface fire regimes, and erosion processes; and (4) fire-triggered increases in watershed runoff and erosion. Two additional illustrations of disturbance interactions, linking forest dieback to erosion processes and fire, are presented below.

Forest dieback interacts with erosion across spatial scales in several ways. For example, after standing dead trees fall to the ground they can become small dams that capture water and sediments. By physically interrupting the connectivity of runoff-producing bare surfaces and channel networks both within and between intercanopy patches, even a few fallen trees can markedly alter hydrological processes and thereby greatly reduce erosion at within-patch and hillslope scales (Figure 1 forked green arrow from dieback to erosion). The death of canopy trees also leads to a decline in

tree competition for water and soil nutrients while allowing more sunlight to reach the ground, facilitating increases in herbaceous surface cover with associated feedbacks to reduce runoff and erosion.

As a second example of disturbance interactions, landscape-scale forest mortality obviously alters fuel structures and thus fire regimes (Figure 1 linked green and red arrows from dieback to fire). However, it is more complicated than “dead trees increase fire risk”, as the feedbacks between climate-induced forest mortality and fire behavior are variable and not well characterized. Once woody vegetation dies it rapidly loses aerial fine fuels (by shedding needles and leaves) and associated volatile (and flammable) biochemical compounds (like aromatic terpenes). This creates standing woody fuel structures that no longer sustain running crown fires, as standing coarse woody fuels do not burn explosively, so risks of unsuppressable crown fires may decrease initially. Conversely, surface fine fuels generally increase rapidly once forest canopies open up from dieback, due to leaf drop and increased herbaceous growth, increasing short-term surface fire probabilities. As time passes the dead trees fall and woody vegetation regenerates, creating complex and heavy surface and ladder fuel loads, and risk of higher severity fire increases again. Thus disturbance-mediated temporal transitions in ecosystem patterns (for example, post-fire vegetation recovery and succession) can also feed back to change disturbance regime dynamics.

CONCLUSIONS

Forest dieback, fire regime changes, and erosion processes in northern New Mexico landscapes provide spatially explicit examples of how pattern-process relationships and disturbance interactions across spatial scales can drive major ecosystem transitions. In particular, changes in the spatial patterns of ecosystem components at intermediate spatial scales (for example, the connectivity of surface vegetation cover) have triggered nonlinear transitions in disturbance processes (for example, fire regimes, erosion) at broader spatial scales (Figures 1, 7), with resultant feedbacks to the ecological patterns at multiple spatial scales (for example, further changes in vegetation cover). Process-based understanding of such positive feedback loops and nonlinear thresholds in disturbance processes across spatial scales are poorly quantified, significantly constraining cross-scale predictions of changes in ecological patterns and processes. In addition, although our knowledge of synergistic interactions among disturbance pro-

cesses across spatial scales remains rudimentary, we know that such cross-scale interactions can be key drivers of ecosystem dynamics, and that threshold-related surprises should be expected, as illustrated by recent landscape changes in northern New Mexico. The pervasive, ongoing magnification of human effects on the planetary ecosystem at all spatial scales highlights the need to improve our conceptual and quantitative understandings of cross-scale interactions among ecological disturbances, to better predict and more sustainably manage our impacts at scales ranging from local to global.

ACKNOWLEDGEMENTS

This work was supported by the Global Change Program of the U.S. Geological Survey's Biological Resources Discipline, the USGS Fort Collins Science Center, and the USDI National Park Service, Bandelier National Monument. I particularly thank Kay Beeley (Bandelier National Monument) and Rebecca Oertel (USGS) for data collection and figure support, Tom Swetnam's fire history group at the University of Arizona Laboratory of Tree-Ring Research for many collaborations through the years, Loren Iverson and Janine Paschke for West-wide GIS data from the US Forest Service's Aerial Detection Forest Health Monitoring program, Dave Breshears (University of Arizona) and Brad Wilcox (Texas A&M University) for long-standing ecohydrology collaborations, and Deb Peters (USDA Jornada Experimental Range) for helpful discussions on cross-scale interactions. Comments by Nate Stephenson, Don McKenzie, Matt Brooks, Julio Betancourt, Susana Bautista, Dave Breshears, an anonymous reviewer, and especially Dean Urban improved this paper.

REFERENCES

- Allen CD. (1989) Changes in the landscape of the Jemez Mountains, New Mexico. Ph.D. dissertation, Berkeley: University of California. 346p.
- Allen CD. 2002. Lots of lightning and plenty of people: an ecological history of fire in the upland Southwest. In: Vale TR, Ed. *Fire, native peoples, and the natural landscape*. Covelo: Island Press. pp 143–93.
- Allen CD, Breshears DD. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proc Natl Acad Sci USA* 95:14839–42.
- Allen CD, Savage M, Falk DA, Suckling KF, Swetnam TW, Schulke T, Stacey PB, Morgan P, Hoffman M, Klingel J. 2002. Ecological restoration of Southwestern ponderosa pine ecosystems: a broad perspective. *Ecol Appl* 12(5):1418–33.
- Berryman A, Stenseth ANC, Wollkind DJ. 1982. Metastability of forest ecosystems infested by bark beetles. *Res Popul Ecol* 26:13–29.

- Bodin O, Elmquist T, Tengo MJ, Norman A, Lundberg J, Elmgvist T. 2006. The value of small size: loss of forest patches and ecological thresholds in southern Madagascar. *Ecol Appl* 16:440–51.
- Breshears DD, Allen CD. 2002. The importance of rapid, disturbance-induced losses in carbon management and sequestration. *Glob Ecol Biogeogr Lett* 11:1–15.
- Breshears DD, Cobb NS, Rich PM, Price KP, Allen CD, Balice RG, Romme WH, Kastens JH, Floyd ML, Belnap J, Anderson JJ, Myers OB, Meyer CW. 2005. Regional vegetation die-off in response to global-change type drought. *Proc Natl Acad Sci USA* 102:15144–48.
- Brooks ML, Matchett JR. 2006. Spatial and temporal patterns of wildfire in the Mojave Desert: 1980–2004. *J Arid Environ* 67:148–64.
- Brooks ML, D'Antonio CM, Richardson DM, DiTomaso JM, Grace JB, Hobbs RJ, Keeley JE, Pellant M, Pyke D. 2004. Effects of invasive alien plants on fire regimes. *Bioscience* 54:677–88.
- Burkett VR, Wilcox DAR, Barrow W, Fagre D, Baron J, Price J, Nielsen JL, Allen CD, Peterson DL, Ruggerone G, Doyle T. 2005. Nonlinear dynamics in ecosystem response to climatic change: case studies and policy implications. *Biocomplexity* 2:357–94.
- Cram DS, Baker TT, Boren JC (2006) Wildland fire effects in silviculturally treated vs. untreated stands of New Mexico and Arizona. Fort Collins: USDA Forest Service Research Paper RMRS-RP-55. 28p.
- Davenport DW, Breshears DD, Wilcox BP, Allen CD. 1998. Sustainability of piñon-juniper woodlands—a unifying perspective of soil erosion thresholds. *J Range Manage* 51:231–40.
- D'Antonio CM, Vitousek PM. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annu Rev Ecol Syst* 23:63–87.
- Esque TC, Schwalbe CR. 2002. Alien annual grasses and their relationship to fire and biotic change in Sonoran Desertscrub. In: Tellman B, Eds. *Arizona-Sonora Desert Museum Studies in Natural History*. Tucson: University of Arizona Press and Arizona-Sonora Desert Museum. pp 165–95.
- Fensham RJ, Holman JE. 1999. Temporal and spatial patterns in drought-related tree dieback in Australian savanna. *J Appl Ecol* 36:1035–60.
- Finney MA. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. *For Sci* 47:219–28.
- Finney MA, McHugh CW, Grenfell IC. 2006. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Can J For Res* 35:1714–22.
- Franklin JF, Shugart HH, Harmon ME. 1987. Tree death as an ecological process. *Bioscience* 37:550–56.
- Gitlin AR, Stultz CM, Bowker MA, Stumpf S, Ecton K, Kennedy K, Monoz A, Bailey JK, Whitham TG. 2006. Mortality gradients within and among dominant plant populations as barometers of ecosystem change during extreme drought. *Conserv Biol* 20:1477–86.
- Gonzalez P. 2001. Desertification and a shift of forest species in the West African Sahel. *Clim Res* 17:217–28.
- Gunderson L, Holling CS. 2002. *Panarchy: understanding transformations in human and natural systems*. Washington: Island Press.
- IPCC (2001) Climate change 2001: synthesis report. A contribution of working groups I, II, and III to the 3rd Assessment Report of the Intergovernmental Panel on Climate Change. In: Watson RR and the Core Writing Team (eds). Cambridge: Cambridge University Press. 398 pp.
- Johansen MP, Hakonson TE, Breshears DD. 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. *Hydrol Process* 15:2953–65.
- Kitzberger T, Brown PM, Heyerdahl EK, Swetnam TW, Veblen TT. 2007. Contingent Pacific–Atlantic Ocean influence on multicentury wildfire synchrony over western North America. *Proc Natl Acad Sci USA* 104:543–48.
- Knapp PA. 1998. Spatio-temporal patterns of large grassland fires in the intermountain West, U. S. A. *Glob Ecol Biogeogr Lett* 7:259–72.
- Logan JA, Powell JA. 2001. Ghost forests, global warming, and the mountain pine beetle. *Am Entom* 47:160–72.
- Ludwig D, Jones D, Holling CS. 1978. Qualitative analysis of insect outbreak systems: the spruce budworm and the forest. *J Anim Ecol* 47:315–32.
- McAuliffe JR, Scuderi LA, McFadden LD. 2006. Tree-ring record of hillslope erosion and valley floor dynamics: Landscape responses to climate variation during the last 400yr in the Colorado Plateau, northeastern Arizona. *Glob Planet Change* 50:184–201.
- Miller C, Urban DL. 2000. Connectivity of forest fuels and surface fire regimes. *Landsc Ecol* 15:145–54.
- Mirit MN, Rodriguez-Buritica S, Wright SJ, Howe HF. 2007. Episodic death across species of desert shrubs. *Ecology* 88:32–6.
- Moore MM, Covington WW, Fule PZ. 1999. Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. *Ecol Appl* 9:1266–77.
- Mueller RC, Scudder CM, Porter ME, Trotter III RT, Gehring CA, Whitham TG. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. *J Ecol* 93:1085–93.
- Nepstad DG, Carvalho G, Barros AC, Alencar AC, Capobianco JP, Bishop, Moutinho P, Lefebvre P, Silva UL Jr, Prins E. 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. *For Ecol Manage* 154:395–407.
- Pedersen BS. 1998. The role of stress in the mortality of Midwestern oaks as indicated by growth prior to death. *Ecology* 79:79–93.
- Peters DPC, Pielke RA Sr, Bestelmeyer BT, Allen CD, Munson-McGee S, Havstad KM. 2004. Cross scale interactions, nonlinearities, and forecasting catastrophic events. *Proc Natl Acad Sci USA* 101:15130–5.
- Peters DPC, Bestelmeyer BT, Herrick JE, Fredrickson EL, Monger HC, Havstad KM. 2006. Disentangling complex landscapes: new insights to forecasting arid and semiarid ecosystem dynamics. *BioScience* 56:491–501.
- Peters DPC, Bestelmeyer BT, Turner MG (2007) Cross-scale interactions and changing pattern-process relationships: consequences for system dynamics. *Ecosystems* (this issue).
- Powell JA, Logan JA. 2005. Insect seasonality: circle map analysis of temperature-driven life cycles. *Theor Popul Biol* 67:161–79.
- Rothman HK. 1992. *On rims and ridges: the Los Alamos area since 1880*. Lincoln: University of Nebraska Press, p 376.

- Sanfranyik L, Carroll AL (2006) The biology and epidemiology of the mountain pine beetle in lodgepole pine forests. In: Sanfranyik L, Wilson B eds. *The Mountain Pine Beetle: A Synthesis of its Biology, Management and Impacts on Lodgepole Pine*. Victoria: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre. pp 3–66.
- Savage M, Mast JN. 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Can J For Res* 35:967–77.
- Schlesinger WH, Reynolds JF, Cunningham GL, Huenneke LF, Jarrell WM, Virginia RA, Whitford WG. 1990. Biological feedbacks in global desertification. *Science* 247:1043–48.
- Seager R, Ting MF, Held IM, Kushnir Y, Lu J, Vecchi G, Huang H-P, Harnik N, Leetmaa A, Lau N-C, Li C, Velez J, Naik N. 2007. Model projections of an imminent transition to a more arid climate in Southwestern North America. *Science* 316:1181–1184.
- Shakesby RA, Doerr SH. 2006. Wildfire as a hydrological and geomorphological agent. *Earth Sci Rev* 74:269–307.
- Shaw JD, Steed BE, DeBlander LT. 2005. Forest inventory and analysis (FIA) annual inventory answers the question: what is happening to our pinyon-juniper woodlands? *J For* 103(6):280–5.
- Sisk TD, Savage M, Falk D, Allen CD, Muldavin E, McCarthy P. 2005. A landscape perspective for forest restoration. *J For* 103(6):319–20.
- Suarez ML, Ghermandi L, Kitzberger T. 2004. Factors predisposing episodic drought-induced tree mortality in *Nothofagus*-site, climatic sensitivity and growth trends. *J Ecol* 92:954–66.
- Swetnam TW, Baisan CH (1996) Historical fire regime patterns in the southwestern United States since AD 1700. Allen CD, technical editor. *Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium*. Fort Collins: USDA Forest Service General Technical Report RM-GTR-286. pp 11–32.
- Swetnam, TW, Baisan CH (2003) Tree-ring reconstructions of fire and climate history in the Sierra Nevada and Southwestern United States. In: Veblen TT, Baker W, Montenegro G, and Swetnam TW eds. *Fire and climatic change in temperate ecosystems of the Western Americas*. Ecological Studies, vol 160, New York: Springer. pp 158–95.
- Swetnam TW, Betancourt JL. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. *J Clim* 11:3128–47.
- Swetnam TW, Allen CD, Betancourt JL. 1999. Applied historical ecology: Using the past to manage for the future. *Ecol Appl* 9:1189–1206.
- Touchan R, Allen CD, Swetnam TW (1996) Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. Allen CD, technical editor. *Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium*. Fort Collins, CO: USDA Forest Service General Technical Report RM-GTR-286. pp 33–46.
- Turner MG. 2005. Landscape ecology in North America: past, present, and future. *Ecology* 86:1967–74.
- Turner MG, Gardner RH, O'Neill RV. 2001. *Landscape ecology in theory and practice: pattern and process*. New York: Springer, p 404.
- US Forest Service (2003) Forest insect and disease conditions in the Southwestern Region. Publication No. R3-04-02. Albuquerque: US Forest Service, Southwestern Region, NM. 34p.
- Veenhuis J (2002) Effects of wildfire on the hydrology of Capulin and Rito de los Frijoles Canyons, Bandelier National Monument, New Mexico. Albuquerque: US Geological Survey, Water Resources Investigations Report 02-4152.
- Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW. 2006. Warming and earlier spring increases western U.S. fire activity. *Science* 313:940–43.
- Whelan RJ. 1995. *The ecology of fire*. Cambridge: Cambridge University Press, p 346.
- Wilcox BP, Breshears DD, Allen CD. 2003. Ecohydrology of a resource-conserving semiarid woodland: effects of scale and disturbance. *Ecol Monogr* 73:223–39.
- Wootton EO (1908) The range problem in New Mexico. Agriculture Experiment Station Bulletin No. 66. Las Cruces: New Mexico College of Agriculture and Mechanic Arts. 46p.